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Towards life cycle sustainability assessment: drawing on the NEEDS project's total cost and multi-criteria decision analysis ranking methods

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Abstract
Purpose In the European Union project New Energy Externalities Development for Sustainability (NEEDS), power generation technologies were ranked by means of two sustainability assessment approaches. The total costs approach, adding private and external costs, and a multi-criteria decision analysis (MCDA) were used, integrating social, economic and environmental criteria. Both approaches relied on environmental indicators based on life cycle assessment. This study aims to analyse the extent to which the development of life cycle sustainability assessment (LCSA) can draw on these ranking methods.
Methods The approaches to rank technologies in the NEEDS project are reviewed in terms of similarities and differences in concept, quantification and scope. Identified issues are discussed and set into perspective for the development of a potential future LCSA framework.
Results and discussion The NEEDS MCDA and total costs considerably overlap regarding issues covered, except for several social aspects. Beyond total costs being limited to private and external costs, most notable conceptual differences concern the coverage of pecuniary (i.e. price change-induced) external effects, and potential double-counting for instance of resource depletion or specific cost components. External costs take account of the specific utility changes of those affected, requiring a rather high level of spatial and temporal detail. This allows addressing intra- and inter-generational aspects. Differences between both ranking methods and current LCSA methods concern the way weighting is performed, the social aspects covered and the classification of indicators according to the three sustainability dimensions. The methods differ in the way waste, accidents or intended impacts are taken into account. An issue regarding the definition of truly comparable products has also been identified (e.g. power plants).
Conclusions For the development of LCSA, the study suggests that taking a consequential approach allows assessing pecuniary effects and repercussions of adaptation measures, relevant for a sustainability context, and that developing a life cycle impact assessment for life cycle costing would provide valuable information. The study concludes with raising a few questions and providing some suggestions regarding the development of a consistent framework for LCSA: whether the analyses in LCSA shall be distinguished into the three dimensions of sustainable development at the inventory or the impact level also with the aim to avoid double-counting, whether or not LCSA will address exceptional events, whether or not benefits shall be accounted for and how to deal with methodological and value choices (e.g. through sensitivity analyses).

Keywords Multi-criteria decision analysis · External costs · Technology ranking · Life cycle sustainability assessment

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1 Introduction

Life cycle assessment (LCA) aims at comparing different products (goods and services) regarding their environmental impacts. Even though staying within environmental limits is

important, sustainability as a concept additionally encompasses human development with both inter-generational and intra-generational considerations. Life cycle sustainability assessment (LCSA) therefore enlarges the scope by integrating additional social and economic aspects into the decision making process with the aim to have more sustainable products (Kloepffer 2008; Heijungs et al. 2010; UNEP/SETAC LCIn 2011). Yet, a common consistent framework to integrate the three components of LCSA, i.e. conventional LCA, social LCA (SLCA) and life cycle costing (LCC), is not yet available and only a few attempts of such an integrated assessment have been made so far (Kloepffer 2008; UNEP/SETAC LCIn 2011).

In search for life cycle-based comparative sustainability assessments, two integrated sustainability assessment approaches were applied to the case of future electricity production in the New Energy Externalities Development for Sustainability (NEEDS) project[1] funded by the European Union (EU). The total costs approach, adding private and external costs, and a multi-criteria decision analysis (MCDA) drew partly on the same methods and data such as elements from LCA, private costs and the impact pathway approach (IPA) used to quantify human health impacts and possible subsequent monetisation (European Commission 2005, as indicated in Fig. S1 in the Electronic Supplementary Material). Elements from MCDA, total cost assessments and LCA have been integrated before. Assessing external costs can be regarded as a special case of an LCA in that human interventions are identified leading to quantified impacts that are then weighted in monetary terms (Krewitt et al. 1998, 2001; Kosugi et al. 2009; Mizsey et al. 2009). MCDAs can be used during weighting in LCAs, for instance, through asking people for their preferences regarding different impact categories (e.g. Soares et al. 2006; Goedkoop and Spriensma 2001b; Rogers and Seager 2009). External costs have already been assessed based on life cycle inventory (LCI) data (cf. NEEDS RS1a partnership 2009) and with help of life cycle impact assessment (LCIA) models (cf. Ott et al. 2006). As a result, taking a life cycle perspective is possible via MCDA and total cost assessments.

In the end, both sustainability assessments from NEEDS yielded rather different technology rankings (cf. Schenler et al. 2009). Beyond aspects such as differing degrees of optimism regarding future private cost developments of different technologies, the algorithm chosen to arrive at a ranking (e.g. dominating alternative or weighted sum, cf. Schenler et al. 2009) and representativeness of the MCDA survey, more fundamentally the differences are due to diverging concepts and scopes of these methodologies. For instance, disregarding a variety of social aspects in the quantified external costs was said to be a main reason why nuclear power ranked rather differently (Ricci 2009).

This study aims to analyse the extent to which the development of LCSA can draw on the ranking methods employed in the NEEDS project. Total costs are similar in concept to societal LCC (Hunkeler et al. 2008) or environmental LCC from a policy-maker perspective (Swarr et al. 2011b). A particular question, therefore, concerns the extent to which the aspects covered by the MCDA can be integrated into the total costs. To this end, conceptual differences between these approaches in terms of scope, i.e. aspects or indicators covered and underlying assumptions, and how the different aspects are balanced against each other, i.e. through weighting, will be explored.

Limiting the scope to methodological questions means that the specific objects of evaluation (i.e. different types of power generation techniques) only matter insofar as they need to fulfil the same function and are characterised in an equally comprehensive way. In the NEEDS project, the kilowatt hour of electricity produced was used as the functional unit as is common when comparing electricity generation techniques; they were characterised by the indicators discussed below.

For the subsequent analysis, indicator coverage and underlying assumptions as well as how to deal with aggregation and weighting within LC(S)A are briefly recapitulated (Section 1.1). Corresponding elements in MCDA and total costs assessments in general and in the NEEDS project in particular are outlined (Sections 1.2 and 1.3).

1.1 LCSA

According to Kloepffer (2008), LCSA integrates conventional LCA with SLCA and LCC. Even if the ISO norms do not exclude assessing social and economic aspects, a consistent framework for LCSA does not yet exist, noting that first proposals were made (Kloepffer 2008; UNEP/SETAC LCIn 2011). All three components include a goal and scope definition, an LCI phase and an interpretation phase (ISO 14040:2006). By contrast, LCIA has not yet been described for LCC (Swarr et al. 2011a). Which impact categories are considered in an LCIA and how depends on the scope definition (ISO 14040:2006; see Sections 3.4 and 3.7 for discussions regarding SLCA and LCC, respectively). In this regard, ISO (14044:2006) gives recommendations concerning international acceptance, minimization of value choices and assumptions, avoiding double-counting, scientific and technical validity of characterization methods and relevance. Regarding the latter, impacts can be assessed at midpoint or endpoint level (e.g. Jolliet et al. 2004). Midpoint indicators are situated within the cause effect-chain while endpoints are situated closer to the final effect on so-called areas of protection such as human health and

[1] www.needs-project.org

natural environment (Udo de Haes and Lindeijer 2002). Furthermore, environmental LCAs are distinguished into attributional and consequential studies (Finnveden et al. 2009). Extending this to LCSA means that the former focuses on all sustainability-relevant flows of the considered life cycle and its subsystems while the latter covers direct or indirect changes resulting from substitutional effects.

According to ISO (14044:2006), weighting is optional in LCIA. It is characterised by the use of numerical factors that are based on value choices. To prevent any misuse, ISO emphasises that weighting only seeks to add information to an LCA while maintaining (i.e. not replacing) other information. Given that weighted results can vary substantially, ISO suggests using several weighting methods. Related to value choices, usually no difference is made in LCSA between impacts occurring at different points in time (cf. Section 3.5).

1.2 Total costs

The total cost approach combines privately borne costs of a certain activity, such as electricity production, with those that are external to that activity (cf. Hirschberg et al. 2004 and Fig. S1, Electronic Supplementary Material). Societal LCC (Hunkeler et al. 2008) or environmental LCC from a policy-maker perspective (Swarr et al. 2011b) are similar in concept, provided the total costs also comprise end-of-life costs. Both building on private and external costs, the total cost concept is related to cost–benefit analysis (CBA). All costs are expressed in the same currency in a given base year (e.g. 2005), allowing to aggregate and compare different cost components. Further names include social, true or full costs.

Beyond the privately borne costs, additional aspects covered are limited to what can be defined as an externality or, when expressed in monetary terms, an external cost. According to the EU-funded Externalities of Energy (ExternE) project series, "an external cost arises, when the social or economic activities of one group of persons have an impact on another group and when that impact is not fully accounted, or compensated for, by the first group" (European Commission 2005, p. 9). Two different kinds of externalities exist (Pearce 1992; Baumol and Oates 1988). *Technological* (or non-pecuniary or true) externalities result from interdependencies between production (e.g. leading to emissions) and utility functions (e.g. reflecting human health risks due to air pollution). All emissions-related externalities, for instance, belong to this category. *Pecuniary* (or pseudo) externalities result from a change in the prices of some inputs or outputs in the economy. Not necessarily producing a misallocation of resources, pecuniary externalities are regularly disregarded in external cost assessments and in CBAs.

There is controversy whether or not deliberate or intended actions undertaken to affect another person's welfare (such as terror risks) are externalities (for more information, refer to the Electronic Supplementary Material). For the purpose of the development of LCSA, consequences of deliberate or intended actions are irrelevant given that the mainly intended area of application is the assessment of products (Kloepffer 2008; UNEP/SETAC LCIn 2011) and associated (economically driven) processes and activities.

In neo-classical economics, using a non-renewable and appropriable resource such as fossil fuels or minerals is not considered an externality: These resources can be extracted (and thus depleted) in a sustainable way according to the Hartwick (or "invest resource rents") rule (Just et al. 2004), assuming substitutability between natural and man-made capital. Without further discussing this contentious assumption, the total cost approach takes into account technological externalities due to mining-associated land use changes and releases. In addition, resource costs are part of the generation costs.

Private and external costs shall be comprehensive as well as spatially and temporally explicit (Pearce et al. 2006; European Commission 1999). Private and external costs will therefore depend on context. External costs are preferably quantified according to the marginal *damage* cost approach (e.g. UBA 2008) as formalised by the ExternE's IPA: It first quantifies site- and time-specific marginal external impacts at the endpoint level that are then monetized (European Commission 2005; Krewitt et al. 1998, 2001). Monetary valuation should be based on the preferences of the general public and not, for instance, on those of politicians or experts (Just et al. 2004; DCLG 2009; Pearce et al. 2006). Due to the inherent uncertainty in quantifying climate change damages, marginal *abatement* costs, i.e. the cost of the last measure to be implemented that helps achieving a policy-set target (Standard-Price Approach, cf. Baumol and Oates 1988), are nevertheless largely accepted[2] to value greenhouse gas emissions (European Commission 2005; Pearce et al. 1996; DECC 2011).

Making impacts across time (inter-generational equity) and space (intra-generational equity) comparable is done through discounting and potentially equity weighting, respectively (Pearce et al. 2006). Discounting means giving lower weights (so-called discount factors) to future impacts by using one or a set of discount rates. Impact assessment guidelines recommend the use of the same discounting scheme for social costs and benefits (e.g. US EPA 2010; European Commission 2009). Equity weighting accounts

[2] These can, however, not be used in cost–benefit analyses of climate change-related measures, for self-fulfilling prophecy reasons (i.e. costs equalling benefits both being based on the same data).

for the fact that a damage of 1 € or US $ is less severe in a highly developed country than in a developing country.

In the NEEDS project, assessing total costs was not a focus. Nevertheless, two research groups calculated total costs (Preiss et al. 2009; Schenler et al. 2009). Besides private costs, externalities due to various emissions and land use changes were considered in the total costs (NEEDS RS1a partnership 2009; Schenler et al. 2009, for more information, see the Electronic Supplementary Material). Particularly for climate change damages, assumptions regarding discounting and equity weighting are critical. In the total costs presented by Preiss et al. (2009), no equity weighting was performed while a positive discount rate of larger than 1 % was used.[3] As an alternative to the marginal damage cost approach, Preiss et al. (2009) estimated an upper bound value for greenhouse gas emissions according to the marginal abatement cost approach. Private costs were discounted at 5 % (Preiss et al. 2009) or at 6 % (Schenler and Bachmann 2008).

1.3 MCDA

The purpose of MCDA is to serve as an aid to thinking and decision making with regard to a complex or multi-faceted problem (DCLG 2009), as is the case when evaluating the sustainability of a product. Thereby, alternatives (e.g. power production techniques) are ranked according to preferences of individuals concerning specific criteria. Any kind of criterion can be chosen in an MCDA if they meet certain requirements (cf. Burgherr 2005; DCLG 2009; Bauler et al. 2007), regarding scientific soundness (e.g. robustness, verifiability, no redundancy or double-counting), functionality (e.g. relevance, completeness) and pragmatism (e.g. as regards comprehensibility and measurability). Spatial and temporal coverage and resolution will depend on the scope of the specific study. Non-binary qualitative criteria need to be converted into numeric values, involving subjective judgements (Hirschberg et al. 2008).

Weighting is performed by the respondents who express their preferences. It needs to be determined whose preferences count. DCLG (2009) calls the persons to involve "key players" that include but are not limited to stakeholders (i.e. people who have a financial or non-financial involvement in the decision's consequences). Different key players will have differing preferences and perspectives reflected in the weights they assign to the criteria, determining the overall ranking. The way the surveyed people are selected affects the representativeness of the final ranking.

In the NEEDS MCDA, the set of relevant quantitative indicators was identified through stakeholder participation (Schenler et al. 2009), as also suggested by Halog and Manik (2011). The criteria were organised hierarchically (Table 1). At the highest level, the criteria were grouped according to the three pillars of sustainability, i.e. the environmental, techno-economic and social dimensions. Below, up to three further levels existed. Quantitative indicators were only available at the lowest level (see Hirschberg et al. 2008 for more details). Quantification of part of the indicators relied on LCSA methods: several of the MCDA indicators are based on LCI data or LCIA results (IDs 1.*x* and 3.3.1.*x* in Table 1); generation costs can be considered LCC results (ID 2.1.1.1). Human health risks from normal operation have even been quantified according to the IPA (without monetisation, cf. Section 1.2). Weighting was performed through an online survey among invited respondents from several European countries (Schenler et al. 2009). In order to arrive at a ranking, the multi-dimensional space of criteria was mapped into a one-dimensional number through a so-called scalarizing function. Two such functions were used in NEEDS: the Dominating Alternative algorithm as the main function while the weighted sum algorithm was used for sensitivity analysis (Schenler et al. 2009).

2 Methods

The approaches to rank technologies in the NEEDS project were reviewed in terms of similarities and differences in concept, quantification and scope. The indicators considered in the MCDA of the NEEDS project are summarised in Table 1. Only the quantitative MCDA indicators are considered in the following. Table 1 shows the MCDA identifiers (IDs) as used in the NEEDS project to reproduce the criteria hierarchy: Remarks made at higher levels also apply to those below. Given that only two aspects were covered by the total costs but not in the MCDA, notably crop yield losses and impacts on building materials (introduced into Table S1 in the Electronic Supplementary Material under ID 2.2), the analysis focuses on the extent to which the MCDA indicators are or can be considered in the total costs. Issues thus identified are discussed and set into perspective for the development of a potential future LCSA framework.

The NEEDS MCDA considered several accident-related indicators (such as MCDA indicators "Hydrocarbons (spills)", "(Radioactive) Land contamination" and "Accident mortality", cf. IDs 1.3.2.*x* and 3.3.2.*x* in Table 1). By contrast, releases and associated impacts from exceptional events such as accidents are generally not quantified in LCA

[3] The discount rate had been determined based on a model-external component of 1 % and a model-internal component that was variable. As a result, the actual discount rate used cannot be specified here. Note further that two sets of climate change-specific external costs were calculated by Schenler and Bachmann (2008) according to the marginal damage cost approach without being explicit about which kind of equity weighting or which discount rate was used.

Table 1 Overview of the indicators used in the NEEDS MCDA (own compilation mainly based on Schenler et al. (2009) and Hirschberg et al. (2008))

Criterion/indicator Name	Unit	MCDA ID
Environment		1
Resources		1.1
Energy[a]		1.1.1
Fossil fuels	MJ/kWh	1.1.1.1
Uranium	MJ/kWh	1.1.1.2
Minerals[a]		1.1.2
Metal ore	kg(Sb-eq.)/kWh	1.1.2.1
Climate[a]		1.2
Carbon dioxide emissions	kg(CO_2-eq.)/kWh	1.2.1.1
Ecosystems		1.3
Normal operation[a]		1.3.1
Biodiversity (impacts through land use)	PDF m^2a/kWh	1.3.1.1
Ecotoxicity	PDF m^2a/kWh	1.3.1.2
Air pollution (acidification & eutrophication)	PDF m^2a/kWh	1.3.1.3
Severe accidents[b]		1.3.2
Hydrocarbons	t/GWeyr	1.3.2.1
(Radioactive) Land contamination	km^2/GWeyr	1.3.2.2
Waste (to be stored)[a]		1.4
Chemical waste	kg/kWh	1.4.1.1
Radioactive waste	m^3/kWh	1.4.1.2
Economy		2
Customers		2.1
Generation cost	EUR/MWh	2.1.1.1
Society		2.2
Direct jobs	Person-years/GWh	2.2.1.1
Fuel autonomy	Ordinal scale	2.2.1.2
Utility		2.3
Financial		2.3.1
Financing risk	Million EUR, net present value	2.3.1.1
Fuel sensitivity	Factor	2.3.1.2
Construction time	Years	2.3.1.3
Operation		2.3.2
Marginal cost	EUR-cents/kWh	2.3.2.1
Flexibility (of dispatch, i.e. forecast times as well as start-up and shut-down times)	Ordinal scale	2.3.2.2
Availability (planned outages, i.e. maintenance, as well as unplanned outages such as failures or unavailable resources)	Factor	2.3.2.3

Table 1 (continued)

Criterion/indicator Name	Unit	MCDA ID
Social		3
Security		3.1
Political continuity		3.1.1
Secure supply (endangered due to few suppliers in the national market in 2050)	Ordinal scale	3.1.1.1
(Possibility of unavailability of a) Waste repository	Ordinal scale	3.1.1.2
Adaptability (to incorporate technological innovations)	Ordinal scale	3.1.1.3
Political legitimacy		3.2
(Risk of triggering domestic) conflicts	Ordinal scale	3.2.1.1
(Need for public) participation	Ordinal scale	3.2.1.2
Risk		3.3
Normal risk[a]		3.3.1
Mortality	YOLL/kWh	3.3.1.1
Morbidity	DALY/kWh	3.3.1.2
Severe accidents[b]		3.3.2
Accident mortality	Fatalities/GWeyr	3.3.2.1
Maximum fatalities	Fatalities/accident	3.3.2.2
Perceived risk		3.3.3
(Perceived risks for) normal operation	Ordinal scale	3.3.3.1
Perceived accidents[b]	Ordinal scale	3.3.3.2
Terrorism[b]		3.3.4
Terror-potential	Ordinal scale	3.3.4.1
Terror-effects	Expected number of fatalities (expert judgement)	3.3.4.2
Proliferation	Ordinal scale	3.3.4.3
Residential environment		3.4
Landscape (functional and aesthetic impacts)	Ordinal scale	3.4.1.1
Noise	Ordinal scale	3.4.1.2

[a] Based on LCI data or LCIA results

[b] Beyond scope of the current analysis

(cf. Frischknecht et al. 2007; JRC-IES 2010a). Related indicators will therefore not be considered in the following analysis.

3 Results and discussion

Several aspects are covered by both the NEEDS MCDA and total costs (cf. Table S1 in the Electronic

Supplementary Material for more details): Both methods cover environmental impacts on climate and on ecosystems[4] due to normal operation (IDs 1.2.1.1 and 1.3.1.*x*), social impacts on human health due to normal operation (ID 3.3.1.*x*) and economic impacts expressed as generation costs (ID 2.1.1.1), at least partially integrating several further cost components (IDs 2.3.*x*). Residential environment-related aspects (i.e. noise and visual annoyances, for instance, from wind converters or transmission lines, IDs 3.4.*x*) can be included in the total costs but were only partly considered in the NEEDS total costs. Differences between the NEEDS MCDA and total costs concern:

- Total costs, like in societal LCC, are limited to private and external costs while for MCDA and LC(S)A, any relevant indicator with an associated unit can be used to express impacts;
- Resource depletion: While using a non-renewable and appropriable resource such as fossil fuels or minerals is not considered a true externality (cf. Section 1.2), associated private costs were part of the total costs (IDs 1.1.*x*; see also Section 3.1);
- Market-induced (or pecuniary) externalities (cf. Section 3.2);
- Comparability between systems (cf. Section 3.3);
- Several social aspects (cf. Section 3.4);
- Waste: similar to LCA not considering waste as an elementary flow (Doka 2009), its mere existence is not a true externality either, unless it leads to land use changes or releases to the environment outside the repository (IDs 1.4.1.*x* and associated IDs 3.1.1.2 and 3.2.1.1).

As regard the concepts (cf. Section 1), differences between LCSA and total costs or MCDA concern:

- The classification of aspects according to the three pillars of sustainability (cf. Section 3.1);
- The treatment of inter-generational and intra-generational equity (cf. Section 3.5);
- The risk of double-counting (cf. Section 3.6);
- The provision of different cost indicators (cf. Section 3.7);
- The regular inclusion of value choices (cf. Section 3.8).

In the following, these differences will be further discussed in view of the associated implications for the development of LCSA.

[4] To evaluate biodiversity impacts, both methods relied upon the LCIA method Eco-indicator 99 (Goedkoop and Spriensma 2001a, b). It is however argued later in this paper that assessing ecosystem services, i.e. the benefits obtained from nature, provides a more meaningful ecosystem damage-related indicator than assessing the potential disappearance of some target species (see also the Online Resource).

3.1 Classification of aspects according to the three pillars of sustainability

MCDA, total costs and LCSA can cover environmental, social and economic aspects. However, assigning an indicator to one of these three sustainability dimensions is not done consistently. LCA, SLCA and LCC, for instance, do not correspond one to one to the three pillars of sustainable development. This is also partly because (ISO compliant) LCA has an organisational perspective, defining environment as the surrounding up to the global scale of a given activity (cf. the environmental management system series, ISO 14001:2005). By contrast, sustainable development generally takes a societal perspective. For instance, pollution, noise and visual intrusion affect people's health or well-being. As done in the NEEDS MCDA and similar to health effects covered by SLCA (UNEP/SETAC LCIn 2009), these impacts need to be considered social impacts if *social* is understood as "of or relating to people or society". Likewise, it is not evident what the *environmental* problem with mineral and fossil fuel depletion is beyond the fact that their extraction leads to land use changes and releases, an issue raised before (Goedkoop 1995; Heijungs et al. 2010). Their scarcity constitutes an *economic* problem (cf. Section 3.6).

3.2 Market-induced (or pecuniary) externalities

In the tradition of CBAs, total costs are mostly concerned with efficiency and not so much with distributional aspects. While the externalities caused by insecurity of fuel supply (IDs 2.2.1.2 and 3.1.1.1 in Table 1, cf. Arnold et al. 2007) mainly stay within the economic dimension of sustainability (pecuniary externalities), the issue of an induced food corn price increase due to a higher demand for corn-based biofuels, for instance, expands into the social dimension and thus potentially constituting a trade-off of higher societal concern. Considering equity concerns of such price effects, for instance, is possible within CBA (Boardman et al. 2006; Pearce et al. 2006) and has already been suggested for LCSA (cf. Heijungs et al. 2010). To account for such impacts, LCSA requires consequential analyses.

3.3 Comparability of systems

Preiss et al. (2009)[5] expanded the studied systems by including back-up technologies, covering the electricity not produced due to non-available fluctuating power generation techniques. By considering the associated private costs, the NEEDS MCDA criteria availability and (partly) flexibility were integrated into the overall total costs (IDs 2.3.2.3 and 2.3.2.2 in Table 1).

Considering such function-related aspects is also relevant for LCSA, being a relative approach in line with ISO (14040:2006) seeking to compare systems providing the same

[5] In contrast to the total costs presented by Hirschberg (2009)

functional unit. Defining the same functional unit is difficult for systems in which rather different techniques fulfil distinct tasks to provide a common function such as intermodal transport (e.g. train, heavy duty vs. light duty vehicles) and electricity supply networks. For instance, individual power generation *techniques* are usually assessed based on a kilowatt hour of electricity produced. However, these individual techniques are not pure substitutes of one another (e.g. wind vs. nuclear, gas turbine power plants vs. lignite power plants) unless they meet the same electricity demand (Ricci 2009). Thus, the goal is to assess electricity supply *systems* fulfilling the same function either through system expansion (i.e. including properly selected back-up technologies) or through a consequential approach that looks at the direct substitutional effects in the system (e.g. Pehnt et al. 2008 for the case of wind power integration). Contrary to Preiss et al. (2009), the environmental implications of the back-up system would need to be included and attributed to the expanded system (here, the fluctuating power source rather than to the unit providing the reserve capacity). User comfort-related functionalities such as degree of automation and thus time requirements (e.g. wood log vs. wood pellet-based domestic heating systems), however, will be difficult to capture in any of the proposed ways and may need to be included in the social assessment as highlighted before (e.g. UNEP/SETAC 2009).

3.4 Coverage of social aspects

Social aspects were important when ranking different techniques (Schenler et al. 2009). Several social aspects are, however, beyond the scope of total costs, limiting its capability of measuring sustainability (cf. comments provided in Table S1 in the Electronic Supplementary Material).

The NEEDS MCDA covered social aspects that are rather different from those currently treated in SLCA. The social and socioeconomic subcategories currently put forward in SLCA were identified based on international agreements (UNEP/SETAC LCIn 2009). Several of the recently suggested SLCA indicators concern workers only (Jørgensen et al. 2012; Traverso et al. 2012). Aspects at the local or national society level and implications for users had been proposed before (e.g. Jørgensen et al. 2008; Labuschagne and Brent 2006). Many of the NEEDS MCDA social indicators take this point of view, being related to accidents, intended impacts and pecuniary externalities, discussed elsewhere in this paper. Several NEEDS MCDA social indicators address particularities of certain techniques. This is supported by the two-layer SLCA approach by Dreyer et al. (2006), suggesting to distinguish between obligatory, i.e. consensus-driven minimum requirements regarding responsible business, and optional, context-specific social indicators. The creation of (direct) jobs was also considered in the NEEDS MCDA (ID 2.2.1.1 in Table 1) although as part of the economic subset of indicators, consistent with its potential consideration in CBA (Pearce et al. 2006).

In general, MCDA and presumably also LCSA are not limited as to which social indicators to include. However, quantification issues remain (Schenler et al. 2009; UNEP/SETAC LCIn 2009). Aspects related to inter-generational and intra-generational equity will be discussed in the next section.

3.5 Treatment of spatial and temporal aspects

In economics, impacts are assessed in a spatially and temporally resolved way: utility changes of those affected shall be quantified according to the marginal damage cost approach. Integrating rather space- and time-dependent externalities into total costs is challenging, for instance, due to variable background levels in view of non-linear cause–effect relationships. This is particularly the case for total costs of general applicability such as at country level. This is presumably why external costs due to noise and visual annoyances (IDs 3.4.1.*x* in Table 1) were only partly considered in the NEEDS total costs.

LCA has only a limited temporal and spatial resolution until now, while noticing related research efforts (cf. Finnveden et al. 2009; Levasseur et al. 2010). Space-dependent compensation mechanisms have already been taken into account in LCIAs of climate change and water uses (Goedkoop et al. 2009; Bayart et al. 2010). Temporal cutoffs are also regularly used (e.g. for global warming potentials valid for a certain time horizon, cf. Hellweg et al. 2003). For social LCA, considering site-specific and more generic implications is considered important (Dreyer et al. 2006). For societal LCC, Hunkeler et al. (2008) acknowledge the need for a positive discount rate, suggesting values of 0.1 % and below.

Regarding spatial and temporal distribution of impacts, the following developments are suggested for LCSA:

- Integrating costs of adaptation measures into LCC, not only related environmental implications into LCA as suggested by Guinée et al. (2011);
- Reporting the temporal and spatial distribution of impacts to highlight the different generations affected: this risks, however, to excessively increase the amount of results to handle, leading to the next point;
- Carrying out sensitivity analyses, in this case, by employing different discounting and equity weighting schemes to aggregate results (as further discussed in the Electronic Supplementary Material);
- Harmonising the use of discounting in LCC and temporal cutoffs in general (cf. Electronic Supplementary Material).

3.6 Risk of double-counting

When trying to arrive at an integrated LCSA framework, double-counting is an issue. According to the LCC promoted

in UNEP/SETAC LCIn (2011), soon to be internalised external costs and benefits shall be accounted for. There is a risk of double-counting that is however not limited to the case when "environmental damages were monetized in LCC" (Kloepffer 2008, p. 91), as is the case for CO_2 emissions under the European Union Emission Trading System, for instance. There is a clear overlap of considering mineral and fossil fuel depletion in (environmental) LCA, already questioned in Section 3.1, and including costs for such resources in LCC. Keeping information on costs at different life cycle stages separately (e.g. Kloepffer 2008) solves the problem of providing an indicator on mineral and fossil fuel depletion (limited to the resource extraction phase). Whether this information captures all relevant aspects associated with resource depletion is a question that is, for instance, currently pursued in the EU-funded LC-Impact project. According to Vieira et al. (2011), concerns differ by stakeholder: industry representatives and policy makers are more concerned with consequences in the short-term and long-term, respectively. The identified aspects to be covered primarily fall into the economic domain (e.g. availability, effort increase, substitution and societal value). Using resources should therefore not be part of an environmental assessment, except for changes in land or water use and/or concerning the assimilation of releases.

3.7 Provision of different cost indicators

When quantifying total costs, double-counting is avoided by considering specific cost components such as fuel costs only once. In the NEEDS MCDA, by contrast, certain private cost components were used more than once to reflect different concerns about the costing scheme associated with a given alternative, such as consumer prices, financial risks, fuel price sensitivity and marginal cost (IDs 2.1.1.1, 2.3.1.1, 2.3.1.2 and 2.3.2.1, respectively, in Table 1 with more information provided in Table S1 in the Electronic Supplementary Material). This is similar to LCA, for instance, where the same LCI data such as NOx emissions can be assigned to different impact categories such as acidification, eutrophication, tropospheric ozone formation and respiratory inorganic effects. In LCC, by contrast, an impact assessment phase is not envisaged (Swarr et al. 2011a). Once a cost breakdown structure has been established (cf. Swarr et al. 2011b), however, re-arranging different private cost components in LCC is possible as well, providing sensitivities through a kind of classification and characterisation of, for instance, customer costs, financing risks and variable cost (as done in the NEEDS MCDA, cf. Schenler et al. 2009). Similarly, short-term- and long-term extraction costs provide different kinds of information. While adding characterised costs risks to double-count, they provide more sustainability-relevant economic information in an LCSA.

3.8 Value choices

MCDA, total costs and LCSA are intended to and used for informing the decision making process by comparing different alternatives. MCDA and total costs provide a framework for interpreting indicator results in view of a specific decision to be taken (Linkov and Seager 2011). To this end, value judgements or weighting is an integral element. By contrast, LCA does not provide such a framework, unless the optional weighting step is carried out. Even though ISO is very cautious when it comes to weighting, it is needed in one way or the other when trying to arrive at an unambiguous conclusion. This is because LCAs regularly provide results for several impact categories which most of the time will not point into the same direction. With many more indicators provided by an LCSA, this will be even more challenging. As a result, deciding on which of the investigated alternatives performs best involves trade-offs. These need to be addressed either informally through discussion or formally through quantitative weighting methods. Informal discussion opens the door for the investigator to argue in favour of his/her own preferences or it comes with the risk of other biases (cf. Rogers and Seager 2009). Formal weighting methods have the advantage of making value judgements more explicit and potentially more transparent, particularly when not only one such method is applied, but several, adopting different perspectives (such as the hierarchist, egalitarian or individualist, Goedkoop et al. 2009). For the time being, it is the analyst who decides on the way how to address the trade-offs and whether or not to provide results obtained by using different weighting methods.

4 Conclusions

Two integrated sustainability assessments, i.e. total costs and MCDA, as used in the NEEDS project were analysed in view of potential further developments of LCSA. In general, total costs and similarly societal LCC are subject to the strictest methodological requirements. This most notably concerns (1) the use of monetary units, (2) the limitation of effects to true externalities and (3) assessing spatially and temporally explicit effects. Nevertheless, the NEEDS MCDA and total costs considerably overlap regarding issue coverage, except for several social aspects which in turn are rather different from those covered by SLCA.

The analysis has shown that

– A classification of aspects according to the environmental, social and economic domain of sustainability is not as clear cut as the distinction into LCA, LCC and SLCA may suggest. This was shown for health effects and the complex issue of resource depletion (cf. Section 3.1). A

suggestion is made below how to organise LCSA according to the three pillars of sustainability;

- Double-counting is not limited to the case when external costs are considered in LCC, resource depletion being again identified as an issue (cf. Section 3.6);
- Rather than staying at the inventory level, developing an impact assessment for LCC is suggested, providing further economically relevant information (cf. Section 3.7);
- Regularly carrying out consequential LCAs will allow extending the assessment beyond true externalities, through the consideration of so-called pecuniary externalities, i.e. market-induced effects that are not limited to environmental impacts (cf. Section 3.2). If consequences in terms of adaptation or compensation are considered, environmental, social and economic implications should be accounted for in LCSA (cf. Section 3.5). Consequential analyses can also help making alternatives truly comparable (cf. Section 3.3);
- System expansion is useful to arrive at the comparability of systems not only in case of appropriately attributing pressures and impacts of multi-output processes but also for integrated technological systems such as the electricity sector or in logistics with intermodal transport (cf. Section 3.3);
- The total cost approach is rather limited in terms of social aspect covered. While social aspects put forward in the recent life cycle literature focus on workers, the inclusion of social implications at societal level appears recommendable, noticing quantification issues (cf. Section 3.4);
- In contrast to LCA, economic analyses require impacts to be quantified in a temporally and spatially resolved way. In order for LCSA to also explicitly address inter-generational and intra-generational equity questions, distinguishing impacts according to when and where they occur is suggested. Using temporal cut-offs and discounting in LCSA require harmonisation (cf. Section 3.5);
- Unless weighting is carried out, LCSA does not provide a framework for interpreting indicator results in view of a specific decision to be taken (cf. Section 3.8), further elaborated upon below.

These findings lead to the following final considerations. Kloepffer's idea (2008) to have three different assessments that are sufficiently compatible to be readily integrated first of all leads to the question: How to separate the different life cycle-based assessments into environmental, economic or social in particular to avoid double-counting? When following the environmental mechanism or, more generally perhaps, the sequence of pressure-state-impact, the three domains could be distinguished either at the level of the inventory/pressure (use of materials, energy, land, water vs. costs vs. social indicators) or at the level of end-points/impacts/areas of protection. If in analogy to ISO (14040:2006), carrying out both, a kind of life cycle sustainability inventory (LCSI) study or an LCSA study shall be possible, a distinction needs to be made at the inventory level. In order to avoid having, for instance, health impacts in both the environmental LCA and SLCA and with the aim to provide more meaningful indicators (i.e. at the endpoint level despite their current level of maturity, cf. JRC-IES 2011), the classification should be at the impact or benefit level. This would require particularly the re-organisation of current environmental LCA.

When distinguishing LCSA at the inventory level, Kloepffer's option 1 formula (2008) can be re-written as:

$$\mathrm{LCSA} = \text{environmental intervention LCA} + \text{economic LCA} + \text{social LCA},$$

noting the question where to account for resource depletion and how.

When distinguished at the endpoint level and acknowledging that the ecosystem services concept is increasingly appreciated by policy makers (e.g. TEEB 2010; European Commission 2011a, b) and by businesses (Hanson et al. 2012; WBCSD 2009) despite remaining quantification issues (see also the Electronic Supplementary Material), one obtains:

$$\mathrm{LCSA} = \text{ecosystem services–related LCA} + \text{economic LCA} + \text{social LCA},$$

the latter including human health impacts in this case.

A question for LCSA concerns the inclusion of risks. At present, releases and associated impacts from exceptional events such as accidents are generally not quantified in LCA. The LCA-related ISO norms, however, do not seem to rule out the consideration of risks. A resulting question is then what the specific cutoff criterion in terms of frequency or likelihood is to classify an event to be disregarded as exceptional or not (accident vs. incident such as oil spills) as done by Frischknecht et al. (2007). Making use of risk assessments in addition (or instead of) conventional LCSA tools has been proposed by Guinée et al. (2011) "for certain sustainability questions", without providing further details. If risks are considered decisive elements in a sustainability context, at least discussing them appears recommendable (JRC-IES 2010b).

According to UNEP/SETAC Life Cycle Initiative (2011), LCSA evaluates "all environmental, social and economic negative impacts and benefits". However, up to now, positive impacts are merely covered through the negative impacts being smaller, two exceptions being the positive climate change and tropospheric ozone formation impacts

(only individualist perspective, cf. De Schryver et al. 2011) and job creation (Labuschagne and Brent 2006). By contrast, external costs regularly include positive environmental impacts (European Commission 2005), even though generally being small. These benefits are determined by means of the damage cost approach in a spatially and temporally specific way, partly considering non-linearities. In LCA, spatially explicit assessments will be increasingly enabled with the advent of ecoinvent version 3 and global, spatially resolved LCIA methods such as Impact World+. Thereby, differences not only in the way of production but also in the vulnerability of the affected environment could be accounted for, a precondition to also quantifying environmental and health benefits. Beyond these benefits, customer comfort aspects such as time requirements or jobs created may be further examples of benefits to be potentially included in LCSA.

Finally, a strong point of total costs and MCDA is that they provide a framework for interpreting results, involving value choices. The ISO norm is concerned about value choices, constituting non-scientific elements (ISO 14044:2006). Doing without them, on the other hand, opens the door for certain biases (see Section 3.8). Not only value choices but also methodological choices made by different groups of individuals will have an influence on the final results (cf. De Schryver et al. 2011; Heijungs et al. 2010). This will also concern the selection of a scenario for future developments of environmental, social and economic conditions, affecting the magnitude of the quantified potential impacts. The question arises how LCSA will deal with these choices. Involving stakeholders has been mentioned (Heijungs et al. 2010). When publicly disclosing an LCSA study, the question is whether the stakeholders involved were sufficiently representative. From a societal perspective, the (informed) general public should be surveyed when eliciting preferences in a CBA (DCLG 2009). Practicality and resource considerations will prevent involving a representative sample of stakeholders in each and every LCSA. Related to this is the question whether it is possible to arrive at a set (or rather several such sets for sensitivity analyses) of methodological and value choices that could be accepted from a societal perspective. Similar to the standard-price approach, the eco-scarcity method, for instance, uses environmental policy targets to derive weights (Frischknecht et al. 2009). In that case, these are, however, only valid for one country. In certain areas of the world, common standards, including parameter values such as discount rates, may exist that are valid for several countries, such as the EU. Given the complexity of life cycles of industrial products and services, however, global standards would be needed in many cases. This will require a globally harmonised effort, potentially to be pursued in the UNEP/SETAC Life Cycle Initiative's third phase. It is expected that such an effort will also come up with different sets of weights and thus the requirement to carry out sensitivity analyses. For the time being, frameworks based on cultural theory, initiated by Hofstetter (1998) and with the latest related publication by De Schryver et al. (2011), may serve as a good basis to provide a formal way of transparently dealing with methodological (i.e. which models, data or scenarios to take) and value choices through sensitivity analyses in LCSA.

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